ELSEVIER

International Biodeterioration & Biodegradation 45 (2000) 151–157

INTERNATIONAL BIODETERIORATION & BIODEGRADATION

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The effects of elk and cattle foraging on the vegetation, birds, and small mammals of the Bridge Creek Wildlife Area, Oregon

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Abstract

High densities of elk (*Cervus elaphus*), especially when combined with cattle (*Bos taurus*), may adversely affect local reforestation efforts and reduce forage availability. Few studies, however, have assessed the potential impacts of high densities of elk, combined with cattle, on biodiversity. We compared vegetation, bird, and small mammal diversity of three elk and cattle exclosures (ungrazed sites) to three grazed sites in the Blue Mountains of eastern Oregon. Shrub species richness was greater on ungrazed than grazed sites ($P = 0.04$). We found no differences in herbaceous vegetative cover, biomass, species richness, or diversity, bird abundance, species richness, or diversity between grazed and ungrazed sites. Small mammal abundance ($P \leq 0.01$), species richness ($P \leq 0.01$), and diversity ($P \leq 0.03$) were greater on ungrazed than grazed sites. In this study, foraging by elk and cattle appears to be reducing shrub and small mammal biodiversity. © 2000 Published by Elsevier Science Ltd.

1. Introduction

Modern forest management has resulted in rapidly increasing elk (*Cervus elaphus*) populations on many western forest lands. Increasing public demands for big game hunting and other recreational opportunities have generated new revenue for many local communities. This has created local economic and political pressures to manage public forests to increase wild ungulate populations (Thomas, 1979; Witmer and deCalesta, 1992). Many of these same lands are also grazed by livestock. Despite decades of research on elk and livestock grazing interactions, little work has focused on how their herbivory may affect the biodiversity of forest ecosystems.

Historically, forest clear-cuts have been used as a management tool to increase wild ungulate populations, enhancing edge effects and forage availability (Thomas, 1979; Witmer et al., 1985). The effects of this human-induced forest fragmentation on elk population have been dramatic. Prior to European settlement of North America, elk existed in relatively low densities over much of the forested west (Sayler and Martin, 1996). The opening of forest canopies through timber harvest has allowed some local populations to expand rapidly, primarily because of increased forage production, reduced predation, and conservative ungulate harvest management. The Blue Mountains of Oregon provide a classic case of the effects of timber harvest on elk population densities. In 1926, numbers of elk in the Blue Mountains were estimated to be about 3660. By 1980, the elk population was estimated to be 58,500 (Irwin et al., 1994), a 16-fold increase in density.

Contemporary evidence suggests that foraging by high densities of deer (*Odocoileus* spp.) and elk can severely affect the abundance, composition, and structure of certain woody and herbaceous plant species (Alverson et al., 1988; Jones et al., 1993; Irwin et al., 1994; Case and Kauffman, 1997). However, the secondary impacts of ungulate foraging on animal biodiversity on western forests are not well documented. Because of changes in vegetation structure and compo-

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0964-8305/00/$ - see front matter © 2000 Published by Elsevier Science Ltd.
PII: S0964-8305(00)00036-6

sition, the effects are believed to be ecologically significant (Best et al., 1979; Adler, 1988). In addition, livestock numbers on western forest and range lands probably exceeded those of wild ungulates in many areas for many decades after settlement, further increasing herbivory pressures on forest ecosystems (Irwin et al., 1994) Therefore, it is critical to determine the effects of large ungulate herbivory on patterns of biological diversity and long-term forest productivity at the landscape level. If adverse effects on forest biodiversity are documented, it is incumbent on state and federal resource management agencies to manage wild ungulate populations, primarily through harvest regulations, and livestock grazing allotments so as to avoid those adverse effects (Witmer and deCalesta, 1992).

Irwin et al. (1994) reported that large herbivores have significantly impacted the plants and soils in and near the Blue Mountains eco-region. The impacts include reduction in understory vegetation, growth of seedlings, and soil fertility. Specific effects of herbivory by large mammals may include greatly modifying understory vegetation, altering conifer dominance, reducing litter depth and compacting soils, and possibly, altering nutrient cycling and energy flows in forest ecosystems (Irwin et al., 1994; Frank and Evans, 1997; Frank and Groffman, 1998). Herbivory effects were especially acute in dry regions or drought conditions (Frank and McNaughton, 1992).

Studies in the west also have documented some effects of wild ungulate and livestock foraging on birds and small mammals (Casey and Hein, 1983; Bock et al., 1984; Medin and Clary, 1990). Medin and Clary (1990) found that some bird species preferred ungrazed sites over grazed sites in Idaho. In addition, small mammal species richness and diversity were higher on ungrazed sites.

Currently, elk numbers in the Blue Mountains greatly exceed historical levels (Irwin et al., 1994). In addition, much of this same area has a history of overgrazing by livestock (Melland et al., 1985). We compared measures of abundance and biological diversity of vegetation, birds, and small mammals under different historical and current grazing intensities by elk and cattle (*Bos taurus*) livestock. We hypothesized that heavy grazing pressure by high densities of elk, combined with livestock grazing, could alter vegetation structure and composition and, hence, it would also affect the abundance and diversity of birds and small mammals using that habitat.

2. Materials and methods

2.1. Study site

The study area was located on the 6156 ha Bridge Creek Wildlife Area, Umatilla County, Oregon. The area is managed by the Oregon Department of Fish and Wildlife. Annual precipitation averages 43 cm, of which 15 cm occurs as rain during the growing season (April–July). The remainder occurs primarily as late fall rains and winter snow. Temperatures range from −50°C to 38°C, with the daily temperature in August averaging 28°C and the daily temperature in January averaging −3°C.

The topography of the area is characterized by undulating plateaus and broad ridgetops cut by steep canyons; elevation ranges from 850 to 1220 m. The soils are derived from the underlying basalt flows and volcanic ash deposits. Two soil types, a moderately deep silt and a shallow stony loam, dominate the area.

The area is primarily a natural grassland, dominated by Idaho fescue (*Festuca idahoensis*), interspersed with forested canyons, draws, and upper elevations. Ponderosa pine (*Pinus ponderosa*) habitat types occupy the deep soiled ridgetops and south-facing slopes of deep canyons. Douglas-fir (*Pseudotsuga menziesii*) vegetation types occupy the steep north-facing slopes of canyons. Western larch (*Larix occidentalis*) is an abundant tree species in the Douglas-fir vegetation type.

The history of the Bridge Creek Wildlife Area was documented by Melland et al. (1985). The area was homesteaded by numerous families in the early 1900s. Dryland farming and ranching were the primary sources of income for the homesteaders. However, the shallow soils produced low wheat yields, therefore most farming operations were converted to livestock grazing by 1950.

Livestock have grazed the Bridge Creek Wildlife Area for over 80 years. During this time, cattle, sheep (*Ovis aries*), and horses (*Equus caballus*) have used the range. Grazing was very heavy until about 1960, when deteriorated range conditions necessitated a reduction in livestock numbers. Nevertheless, range conditions in the area continued to deteriorate (Melland et al., 1985). The Oregon Game Commission began to acquire the land between 1961–1973 so that it could be managed as elk winter range.

Because of the deteriorated condition of the range, livestock were not grazed between 1961 and 1964. A range inventory in 1964 determined that 17% of the area was in poor, 40% fair, 40% good, and 3% excellent condition (Anderson et al., 1990). As a result, elk use was far below historic levels. To improve range quality for wintering elk, the 1964 Oregon Game Commission established a seven pasture deferred rest-rotation grazing system in which one pasture, the 365 ha Bridge Creek drainage, would exclude livestock and be managed primarily as elk thermal cover and winter forage. The remaining six pastures were managed with lay-down fences to allow easy passage for migrating elk. Approximately 1000 animal-unit-months (AUMs) of livestock grazing are allowed on two pastures during late spring/early summer. In addition, 1000 AUMs are grazed on another two pastures during the late summer. The remaining two pastures are not grazed by livestock for an entire year. Using this system, each pasture is allowed to rest every third year. Results from a range inventory in 1988 showed that most of the area was in either good or excellent condition. Wintering elk have increased from a low of 214 AUMs in 1963 to over 2000 AUMs by 1995.

Three exclosures were erected between 1982 and 1991 to provide protection for natural regeneration of ponderosa pine, as well as for artificially planted seedlings. Past plantings of ponderosa pine seedlings to increase thermal and security cover for elk in unprotected areas failed because of heavy browsing by wintering elk (Melland et al., 1985).

We used the three elk and cattle exclosures (ungrazed sites) on the Bridge Creek Wildlife Area for this study. The exclosures ranged from 20 to 40 ha in size. All had been in place and allowed to recover from grazing for at least 6 years. The exclosures were designed to exclude both elk and cattle, however, a small amount of cattle grazing was allowed in one exclosure during the summer months of some years. The exclosures occur on *Pinus ponderosa/Symphoricarpos albus* vegetation types.

A grazed site within 300 m of each exclosure was selected for comparison with the ungrazed sites. The grazed sites were of similar historic vegetation and grazing, topography, aspect, and size as its nearby ungrazed sites (Melland et al., 1985).

2.2. Vegetation and organic litter sampling

Vegetation was sampled at 10 sampling stations placed at 20 m intervals along four randomly placed 200 m transects within each site to determine percent age cover, composition, and diversity of plants and organic matter. Herbaceous plant cover and organic litter cover were visually estimated to the nearest percent age using a 20×50 cm plot frame (Daubenmire, 1959). Herbaceous biomass was determined by clipping, bagging, drying (24 h at 100°F), and weighing vegetation from three randomly selected 20×50 cm plots per transect.

We used the line-intercept method to determine horizontal shrub cover. We defined shrub as a woody perennial less than 2 m in height, or greater than 2 m in height but with multiple stems. This definition was intended to include tree species that were less than 2 m in height because they function essentially as a shrub to bird and small mammal use (MacArthur and MacArthur, 1961; MacArthur, 1964). A 30 m tape was placed perpendicular to the 200 m transect at each station. Percent coverage of each shrub species was determined by the amount of crown intercepted by the tape, divided by the total length of the tape, multiplied by 100. The total percent shrub coverage was obtained by adding across shrub species.

2.3. Bird surveys

We surveyed avian species during the months of May and June using fixed-radius point counts (Ralph et al., 1993). Counts were repeated four times per season for each site. Data were averaged over the four counts. Five evenly-spaced points were permanently located in each study site, with points placed 150 m apart to reduce the chance of counting individual birds twice (Ralph et al., 1993). We counted birds from each point for 10 min. Data collected from each point included the species and the number of individuals of each species detected by sight or sound. Only birds observed within 75 m of the point were recorded and used in analysis; birds flying over the point were recorded separately (Ralph et al., 1993). Point-counts began within 15 min of local sunrise, and did not last beyond 0900 h (Ralph et al., 1993). Each grazed site and its nearby ungrazed site were surveyed on the same days to help avoid differences in weather, date, etc., and survey starting points were alternated. Surveys were not conducted when weather interfered with the audibility or visibility of birds.

2.4. Small mammal surveys

We used removal methods to sample small mammals during July and August of 1996. Live-traps would have been used if we had planned to trap repeatedly. We systematically arranged a combination of paired museum special snap-traps and pitfalls (2 kg coffee cans partially filled with water to minimize escape) at 25 trapping stations on four 250 m transects (with 10 m trap-station and transect spacing) per site. One pitfall and one snap-trap were baited with peanut butter and rolled oats, and placed within 1 m of each trapping station. The 100 snap-traps and 100 pitfall traps at each site were operated for five consecutive nights. Traps were set in the evening and checked each morning. Traps were operated on the same nights for each grazed site and its nearby ungrazed site. The fossorial northern pocket gopher (*Thomomys talpoides*) was sampled by counting active burrow systems along the trapping transects. Clusters of fresh dirt mounds were considered to be the actions of a single animal as the northern pocket gopher is very territorial during most of the year (Chase et al., 1982).

2.5. Data analysis

The Shannon–Wiener diversity index (Shannon and

Weaver, 1949) was used to compare vegetation, bird, and small mammal data for grazed and ungrazed sites. The reciprocal of the Simpson diversity index (Simpson, 1949) was also used to compare bird and small mammal data for grazed and ungrazed sites.

Other variables measured and compared included vegetation, bird, and small mammal species richness and abundance. Species richness was the number of species detected during sampling. Vegetation abundance was expressed as percentage canopy cover. Bird abundance was expressed as the number of birds detected during point counts. Small mammal abundance was expressed as the number of individuals captured per 100 trap-nights.

Data were analyzed for the grazed and ungrazed sites using an one-way analysis of variance, and results were considered significant at $P \leq 0.05$. All percentage data were transformed with the arcsine–square root transformation before analyses (Steel et al., 1997).

3. Results

3.1. Vegetation and organic litter

We recorded 98 herbaceous plant species, including 20 grasses, five sedges and rushes, and 73 forbs. Eighty herbaceous species were recorded on ungrazed sites and 76 species on grazed sites. We observed no differences for herbaceous species cover, biomass, species richness, or Shannon–Wiener diversity between grazed and ungrazed sites (Table 1).

We recorded eight shrub species, including seven species on ungrazed sites and five species on grazed sites. Shrub species richness was greater on ungrazed than grazed sites (Table 1). Shrub abundance and Shannon–Wiener diversity were somewhat higher ($P = 0.06$) for ungrazed than grazed sites, suggestive that differences may exist (Table 1).

Although the mean organic litter cover was higher on ungrazed sites (60.93%) than on grazed sites (52.81%), the difference was not significant ($P = 0.18$; Table 1).

Table 1
Abundance, richness, and diversity of vegetation, birds, and small mammals on grazed and ungrazed areas on Bridge Creek Wildlife Area, Oregon, 1996

	Grazed	Ungrazed	*P*-value
Herbaceous vegetation			
Cover (%)	64.60	61.59	0.42
Biomass (kg/ha)	1114.10	1193.30	0.70
Species richness	28.50	27.75	0.63
Shannon–Wiener index	2.54	2.56	0.87
Shrub vegetation			
Cover (%)	2.50	4.40	0.58
Species richness	2.17	3.33	0.04
Shannon–Wiener index	0.33	0.57	0.06
Organic litter			
Cover (%)	52.81	60.93	0.18
Birds			
Abundance (no./point count)	5.92	4.98	0.45
Species richness	2.77	2.85	0.88
Shannon–Wiener index	0.75	0.82	0.70
Simpson's index	2.44	3.15	0.35
Small mammals			
Abundance (no./100 trap-nights)	2.83	7.58	0.00
Species richness	1.33	2.33	0.00
Shannon-Wiener index	0.26	0.64	0.00
Simpson's index	1.26	2.19	0.03

3.2. Birds

We detected 654 birds of 31 species during bird surveys. Two-hundred and ninety-five individuals of 25 species were detected on ungrazed sites and 359 individuals of 23 species were detected on grazed sites. No differences were observed for bird abundance, species richness, Shannon–Wiener diversity, or Simpson diversity between grazed and ungrazed sites (Table 1). Bird abundance tended to be higher on grazed sites; however, species richness and diversity tended to be higher on ungrazed sites (Tables 1 and 2).

3.3. Small mammals

We captured 82 small mammals of three species on both the grazed and ungrazed sites. These include two species of rodents, deer mouse (*Peromyscus maniculatus*) and montane vole (*Microtus montanus*), and one

Table 2
Number of birds observed and small mammals captured on grazed and ungrazed areas on Bridge Creek Wildlife Area, Oregon, 1996

Species	Grazed	Ungrazed
Birds[a]		
American robin (*Turdus migratorius*)	20	25
Brewer's blackbird (*Euphagus cyanocephalus*)	33	41
Chipping sparrow (*Spizella passerina*)	16	29
Dark-eyed junco (*Junco hyemalis*)	19	2
European starling (*Sturnus vulgaris*)	32	9
Mountain chickadee (*Parus gambeli*)	15	2
Red crossbill (*Loxia curvirostra*)	121	53
Western meadowlark (*Sturnella neglecta*)	33	40
Total	289	201
Small mammals		
Deer mouse (*Peromyscus maniculatus*)	18	48
Montane vole (*Microtus montanus*)	1	5
Vagrant shrew (*Sorex vagrans*)	0	10
Northern pocket gopher[b] (*Thomomys talpoides*)	15	30
Total:	34	93

[a] Data presented only for the 8 most common bird species.
[b] Number of active burrow systems observed along transects.

species of insectivore, vagrant shrew (*Sorex vagrans*). In addition, 45 active burrow systems of the northern pocket gopher were recorded. Small mammal abundance, species richness, Shannon–Wiener diversity, and Simpson diversity all were greater ($P \leq 0.03$) on ungrazed than grazed sites (Table 1). Shrews were captured only on ungrazed sites (Table 2). Nearly four times as many deer mice were captured and twice as many northern pocket gopher active burrow systems were detected on ungrazed than grazed sites (Table 2).

4. Discussion

Lack of difference in herbaceous vegetation between grazed and ungrazed sites could be the result of several factors. First, the current livestock grazing system employed by the Oregon Department of Fish and Wildlife is intended to increase herbaceous vegetative biomass for wintering elk (Melland et al., 1985). The merits of using livestock grazing to increase herbaceous plant production have been documented by researchers throughout the western US (Bryant, 1985; Anderson et al., 1990; Sanderson et al., 1990). Increases in plant biomass may be positively related to increases in plant cover, which could explain the lack of significant differences in herbaceous plant cover between grazed and ungrazed sites.

A shift in herbaceous plant species composition caused by grazing would not necessarily reduce plant species richness and diversity. Two communities could have completely different plant communities, but have similar values for both plant species richness and diversity (Magurran, 1988). Green and Kauffman (1995) found that grazed riparian communities in northeastern Oregon could have different plant species composition, but similar values for diversity. Theoretically, elk and cattle grazing might cause a reduction or elimination of some plant species, but those species are replaced by other plant species, resulting in a different species composition but similar values for species richness and diversity.

Although shrubs and tree regeneration were relatively scarce on all sites, mean values for the variables tended to be higher on the ungrazed exclosures. Wintering elk on the area generally browse all above-snow woody vegetation (D.W. Harcombe, Oregon Department of Fish and Wildlife, pers. comm.), thereby reducing shrub cover, species richness, and diversity on all browsed areas. Other studies on western rangelands have reported results of wild ungulate browsing on woody vegetation that were similar to this study (Tiedemann and Berndt, 1972; Edgerton, 1985; Irwin et al., 1994; Case and Kauffman, 1997).

The results of the bird surveys indicated no differences between grazed and ungrazed sites. The high mobility of birds, coupled with relatively small size of the exclosures could be the reasons for the lack of detectable differences. Bird sampling methods were not intended to detect only breeding birds, but all birds using the habitat. Therefore, birds merely passing through the area at the time of the survey were counted, which might have confounded the results. In addition, sample sizes may have been too small to detect real differences in bird populations caused by grazing.

Research on grazing effects on birds has produced variable results. The effects of grazing on bird abundance, species richness, and diversity may depend on the individual species (Casey and Hein, 1983), the habitat type (Rice et al., 1984; Bock et al., 1992), season of grazing (Knopf et al., 1988), and grazing intensity (deCalesta, 1994). The Oregon Department of Fish and Wildlife attempted to evenly distribute livestock through the use of salt-blocks and water developments, yet we did not quantify actual use of each grazed site by cattle. In addition, some grazed sites might have been used more by wintering elk than others. Thus, the uneven distribution of elk and livestock could also have influenced the lack of significant differences in the bird data between grazed and ungrazed sites.

Birds play an important role in ecosystem functioning, contributing to processes such as seed dispersal and energy flow. It is interesting to note that whereas bird abundance tended to be higher on grazed sites, species richness and diversity tended to be lower. Loss of even one or two bird species due to grazing might compromise the integrity of the community. In addition, alteration of habitat due to grazing might be contributing to existing population declines of some migratory bird species (Finch, 1991). Therefore, it is essential to understand and consider the possible negative impacts, large herbivores may be having on bird species in western coniferous forests. Further research is needed to better define these potential relationships.

Somewhat reduced shrub cover and organic litter may have been responsible for the low mean values for small mammal abundance, species richness, and diversity on the grazed sites. Similar studies have also found that grazing by wild ungulates and cattle can reduce the abundance and diversity of small mammals (Crouch, 1981; Medin and Clary, 1989, 1990). Habitat structure (plant species, growth form, organic litter, etc.) is important in determining the composition and densities of small mammal populations (Adler, 1988; Yahner, 1992). Small mammals need vegetative and organic litter cover to protect themselves from both the elements and predation. Kotler et al. (1991) and Longland and Price (1991) found that rodents were more susceptible to predation by owls when they foraged in areas with less cover. Removal of cover by large herbivores could alter the small mammal communities on these sites by increasing their exposure to predators.

Small mammals are important components of ecosystems, and loss of small mammal abundance or diversity might jeopardize ecosystem functions such as nutrient cycling, seed dispersal, herbivory, and predator–prey dynamics (Chew, 1976; French et al., 1976; Potter, 1976). The absence of shrews in grazed areas could indicate the loss of a trophic level on these sites. The deletion of small mammal species as a result of grazing could negatively affect the stability of the community (Elton, 1927; MacArthur, 1955; Frank and McNaughton, 1991; Tilman and Downing, 1994). Further research is needed on this topic to determine the consequences of species deletions in ecosystem functioning.

5. Conclusions

Grazing by high densities of large herbivores can adversely affect ecosystem composition and, presumably, function. We found that foraging by high densities of elk, combined with livestock, reduced shrub cover, species richness, and diversity. This, in turn, may have been associated with lower small mammal abundance, species richness, and diversity. This community alteration might compromise the integrity of the ecosystem, because small mammals are important contributors to several ecosystem processes. Bird populations may not have been adversely affected, although further studies are needed to confirm this finding. The effects of land management practices, including goal densities of wild ungulates and livestock allotments, on low-profile species such as rodents and insectivores should be given more consideration in future planning on public lands in order to properly manage for healthy ecosystems.

Acknowledgements

This study was funded by the USDA National Wildlife Research Center and the Oregon Department of Fish and Wildlife. Rod Sayler, Jack Nelson, Dave Harcombe, Bill Wirth, and Stuart Love were instrumental in the establishment and completion of this study. We thank Marc Evans and Roger Chapman for their valuable statistical support. We also thank our field assistants, Mike Pipas, Janean Creighton, Max Meyer, Andrea Moser, and Rich Huenefeld. Thanks are also due to Tom Thompson, Karen Kronner, and others with the Umatilla National Forest for their input and cooperation on the project.

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